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The *Lancet* Countdown on health benefits from the UK Climate Change Act: a modelling study for Great Britain

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Summary

Background Climate change poses a dangerous and immediate threat to the health of populations in the UK and worldwide. We aimed to model different scenarios to assess the health co-benefits that result from mitigation actions.

Methods In this modelling study, we combined a detailed techno-economic energy systems model (UK TIMES), air pollutant emission inventories, a sophisticated air pollution model (Community Multi-scale Air Quality), and previously published associations between concentrations and health outcomes. We used four scenarios and focused on the air pollution implications from fine particulate matter (PM_{2.5}), nitrogen dioxide (NO₂) and ozone. The four scenarios were baseline, which assumed no further climate actions beyond those already achieved and did not meet the UK's Climate Change Act (at least an 80% reduction in carbon dioxide equivalent emissions by 2050 compared with 1990) target; nuclear power, which met the Climate Change Act target with a limited increase in nuclear power; low-greenhouse gas, which met the Climate Change Act target without any policy constraint on nuclear build; and a constant scenario that held 2011 air pollutant concentrations constant until 2050. We predicted the health and economic impacts from air pollution for the scenarios until 2050, and the inequalities in exposure across different socioeconomic groups.

Findings NO₂ concentrations declined leading to 4 892 000 life-years saved for the nuclear power scenario and 7 178 000 life-years saved for the low-greenhouse gas scenario from 2011 to 2154. However, the associations that we used might overestimate the effects of NO₂ itself. PM_{2.5} concentrations in Great Britain are predicted to decrease between 42% and 44% by 2050 compared with 2011 in the scenarios that met the Climate Change Act targets, especially those from road traffic and off-road machinery. These reductions in PM_{2.5} are tempered by a 2035 peak (and subsequent decline) in biomass (wood burning), and by a large, projected increase in future demand for transport leading to potential increases in non-exhaust particulate matter emissions. The potential use of biomass in poorly controlled technologies to meet the Climate Change Act commitments would represent an important missed opportunity (resulting in 472 000 more life-years lost from PM_{2.5} in the low-greenhouse gas scenario and 1 122 000 more life-years lost in the nuclear power scenario from PM_{2.5} than the baseline scenario). Although substantial overall improvements in absolute amounts of exposure are seen compared with 2011, these outcomes mask the fact that health inequalities seen (in which socioeconomically disadvantaged populations are among the most exposed) are projected to be maintained up to 2050.

Interpretation The modelling infrastructure created will help future researchers explore a wider range of climate policy scenarios, including local, European, and global scenarios. The need to strengthen the links between climate change policy objectives and public health imperatives, and the benefits to societal wellbeing that might result is urgent.

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Introduction

The effects of climate change on human health and wellbeing are many and varied, interacting with a wide range of the social and environmental determinants of health. In the UK, several key short-term risks have been identified, from extremes of weather and the changing patterns of infectious disease (particularly tick-borne disease) to changes in ground-level ozone.¹ The increase in strength and frequency of heatwaves, and of floods, are particularly concerning given their direct effects on health in the form of cardiovascular disease, heat-related mortality, direct injury, and the

mental health effects that result from disruption to lives and livelihoods.¹ These health effects alone provide reason enough for both enhanced mitigation and adaptation activities.

The 2015 *Lancet* Commission on Health and Climate Change²—which assessed the responses to climate change with a view to ensuring the highest attainable standards of health for populations worldwide—concluded that “tackling climate change could be the greatest global health opportunity of the 21st century” and recommended that more accurate national quantification of the health co-benefits and economic

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Research in context

Evidence before this study

We searched the peer-reviewed literature on atmospheric science, energy policy, and climate change, including PubMed. We searched UK Government policy documents on air quality, climate change projections, and policy, including the reports of the UK Climate Change Committee. Our searches covered the period from 1990 to December, 2016. We deemed meta-analyses as not appropriate for this work.

Added value of this study

For the first time, this study has produced a model that links the UK energy system model used by the government and the Climate Change Committee with a sophisticated chemical-transport model for air quality. We then used the resulting concentrations to estimate health impacts and the differential exposures in socioeconomic classes. We used a full approach with concentration, exposure, and concentration-response function to estimate the impacts that represent an

important improvement on the more commonly used damage cost approach, which only uses emissions of air pollutants and not concentrations or exposures.

Implications of all the available evidence

The study has shown the need for careful consideration of the potential disadvantages of policies to reduce greenhouse gas emissions, in particular the downsides for air quality and public health downsides from the promotion of wood-burning in small-scale applications. This study also highlighted the fact that the projected increase in electric vehicles, a keystone of UK policy, will make large reductions in exhaust emissions and substantially reduce urban nitrogen dioxide concentrations, the subject of current High Court cases in the UK. However, the study also indicated that non-exhaust emissions from tyre and brake wear might not decrease.

impacts of mitigation decisions was essential in promoting a low-carbon transition. Building on these foundations, the *Lancet* Countdown: tracking progress on health and climate change³ was formed as an independent research collaboration, dedicated to monitoring the implementation of the Paris Agreement and the health benefits that result through annual publication of a series of indicators.

An important example of these health co-benefits is improvements in air pollution that result from mitigation actions. Poor air quality in the UK has been associated with considerable effects on health—the Committee on the Medical Effects of Air Pollutants (COMEAP) estimated a loss of 340 000 life-years, equivalent to about 29 000 attributable deaths from fine particulate matter (PM_{2.5}) in 2008.⁴ A figure of 6·5 million attributable deaths has been quoted globally.⁵ PM_{2.5} is responsible for most detrimental health effects, with contributions from nitrogen dioxide (NO₂) and ozone. The role of NO₂ is subject to some uncertainty and debate.⁶ Air pollutants have been associated with cardiovascular disease in the form of ischaemic heart disease and stroke, and causing, or exacerbating, existing respiratory diseases such as chronic obstructive pulmonary disorder, asthma, lower respiratory tract infections, and carcinomas of the respiratory tract.⁷

The UK's 2008 Climate Change Act, which requires a reduction of at least 80% in greenhouse gas emissions by 2050, relative to 1990 concentrations, alongside interim carbon budgets agreed up to 2032, represents one of the most ambitious legislative instruments to address climate change mitigation in the world. In principle, attaining the targets under the Climate Change Act could provide large reductions in air pollution emissions. However, the pathways to attaining the target need to be carefully

chosen to avoid unnecessary worsening of air pollution and its attendant health effects, and in turn, to maximise the co-benefits that could arise.⁸ This project has now developed more powerful analytical methods to guide the choice of measures that protect human health by mitigating both climate change and air pollution, while avoiding unintended consequences.

We aimed to supplement the annual indicators presented by the *Lancet* Countdown and support its aims by quantifying the substantial opportunity that exists for improving public health. We looked at the coordination of policies to mitigate climate change and address air pollution simultaneously, bringing together detailed data and advanced methods and models. We also aimed to focus on the health co-benefits that result from mitigation actions.

Methods

Modelling the energy scenarios

The air pollution effects on public health in Great Britain have been quantified, from modelled concentration changes in 2035 and 2050, for a scenario that kept 2011 air pollutant concentrations constant, and for three scenarios generated by the UK TIMES energy system model, which is used by the UK Government and the Committee on Climate Change to explore pathways to the Climate Change Act target.^{9,10} We used these four scenarios in this study. UK TIMES models the UK energy system and produces scenarios, for a given set of economic and technical parameters, for future sectoral energy use in the UK. We used three scenarios: a baseline scenario, which does not meet the Climate Change Act target and envisages no further climate mitigation beyond that already achieved; a nuclear replacement scenario, which meets the Climate Change

Act target and limits nuclear generation capacity to 2015 amounts, resulting in no new builds and 2015 station replacement only; and a low-greenhouse gas scenario, which also meets the Climate Change Act target and which has no policy constraint on nuclear build, only technical and economic feasibility considerations. Regarding the primary energy consumption, both low-greenhouse gas and nuclear power scenarios have much higher concentrations of biomass and biofuels in 2050, an important alternative low carbon source of energy to replace fossil fuels such as gas and oil (figure 1).

Modelling the emission scenarios

We converted the output from the UK TIMES model into an air pollution emission inventory (sulphur dioxide, carbon monoxide, nitrogen oxides [NO_x], volatile organic compounds, PM₁₀, PM_{2.5}, and ammonia) for Great Britain, with a 1km×1km grid resolution, by providing fuel used in petajoules for different industrial or commercial and domestic energy sectors or for road vehicles, in billion vehicles per km. We then linked this conversion to the equivalent sectors of the UK National Atmospheric Emissions Inventory (NAEI).¹¹ Emission factor changes, representing improvements in the emissions' performance within each sector were based on the 2030 NAEI predictions (Misra A, Ricardo-AEA, personal communication), with emission factors remaining constant between 2030 and 2050, except for domestic biomass emission factors, which changed to comply with the European Union's Ecodesign Directive, and vehicles that we assumed to all be Euro 6/VI in 2050, since no plans exist to go beyond this standard. Emission factors remaining at 2030 amounts for other sources is a pragmatic assumption, since creating emission factors for all 570 different source sectors between 2030 and 2050 is a considerable undertaking. The vehicle non-exhaust emission factors are uncertain and as a conservative assumption we have kept them the same for all years.

The UK TIMES model only deals with energy production, and thus emissions from other sources that are important for air pollution need to be added. Emissions of ammonia, largely from agriculture, are responsible for large amounts of secondary inorganic PM_{2.5} and emissions of solvents are important precursors of ozone. We included both emissions of ammonia and of solvents when developing the method for this study, because these emissions include non-exhaust traffic emissions from brake and tyre wear, domestic wood burning, cooking, and diesel vehicle volatile organic compounds, not previously considered in the UK NAEI. Additionally, emissions from Europe are important long-range determinants of air pollution concentrations in the UK. To account for this, we used results from ECLIPSE (version 5a),¹² chosen because the forecast is both recent (from 2016) and includes all the emissions reductions agreed under the European Union member states National Emissions Ceiling Directive.¹³

Modelling air quality in the scenarios

We predicted total air quality in Great Britain using the Community Multi-scale Air Quality (CMAQ) model, coupled to the Atmospheric Dispersion Modelling System roads model, a model described as CMAQ-Urban.^{14–16} CMAQ-Urban outputs provide hourly air pollution concentrations across Europe for every 50 km, every 10 km for the UK, every 2 km in urban areas, and every 20 m close to roads. The most detailed 20 m model was used in London; Bristol, Swansea, and Cardiff; Birmingham; Liverpool and Manchester; Leeds; and Glasgow and Edinburgh, and was evaluated against 80 measurement sites in 2011 and 2012. Meteorological fields were derived from the Weather Research and Forecasting (WRF) model (version 3.6.1).¹⁷ To reflect 2011 and future climate, the WRF model was driven by 2011 and future lateral conditions from the global coupled carbon-climate Earth System Model 2 developed at the National Oceanic and Atmospheric Administration (NOAA) Geophysical Fluid Dynamics Laboratory.¹⁸ The 2011 and future chemical boundary conditions for CMAQ were derived from the global chemical transport model MOZART-4 and the NOAA GFDL-AM3 model.^{19,20}

Predicting the health impacts of future air quality

Modelling the health impacts of climate change mitigation policies requires long-term outlooks, because pollution changes occur up to and beyond 2050. We used the time period 2011–2154 to capture the full effect of the changes on life expectancy, allowing lifetime follow-up after the concentration changes in 2050. The future health impacts across Great Britain were modelled using a life table approach developed by Miller and Hurley²¹ and used in previous reports^{22–25} on the health impacts of air pollution. We averaged the modelled air pollution concentrations in the 10 km rural and 2 km urban grid squares to ward level, and multiplied these concentrations by the ward population, summed to local authority, and

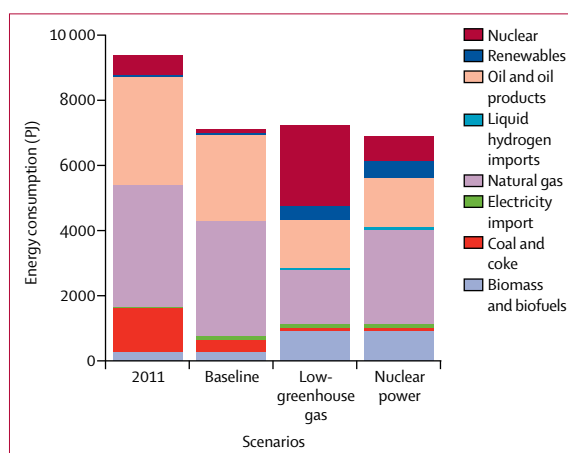


Figure 1: Primary energy consumption in the four scenarios in 2050

Only the low-greenhouse gas and nuclear power scenarios met the UK's Climate Change Act target.

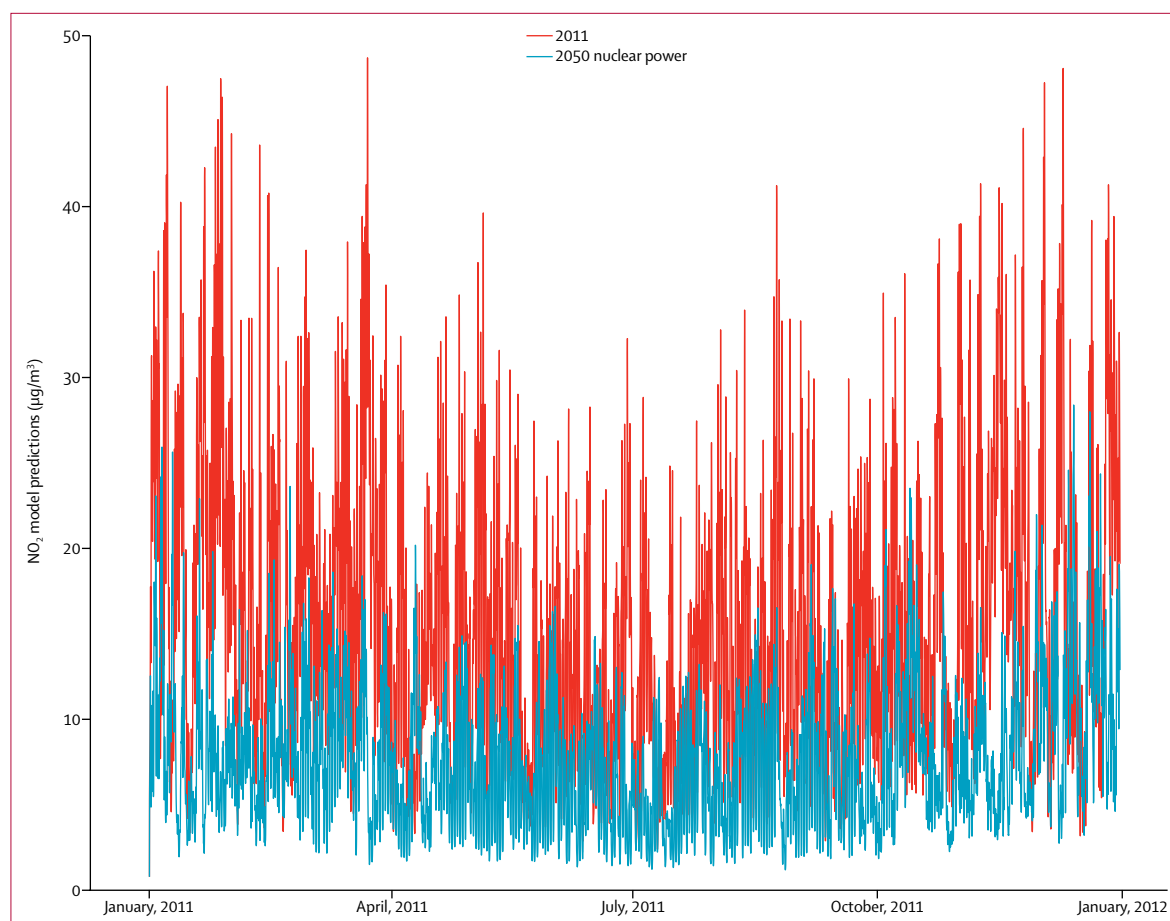


Figure 2: Predicted hourly NO₂ concentrations over the sites in Great Britain at 10 km spatial scale
Data are mean. NO₂=nitrogen dioxide.

divided by the local authority population to give a population-weighted mean for each local authority (population-weighting ward concentrations across the local authority and then calculating health impacts at local authority level is arithmetically equivalent to calculating health impacts for each ward separately). We used these population-weighted concentrations with the concentration-response functions (hazard ratios) and local authority mortality, accounting for new births and mortality improvement projections, to estimate the health outcomes associated with the air pollutants. Accounting for these projections represents a substantial improvement on previous air pollution health impact assessments,^{23–25} and is particularly important given the long-term health impact predictions required for climate change policies.

Ideally, life table analysis uses population and mortality data by 1-year age group. If these data were not publicly available at the relevant geographical scale and year, we inferred them using data from other geographical scales or other years. The starting point was population^{26,27} and mortality (unpublished)²⁸ data for 2011, for each ward by

gender and 5-year age group. For each gender, we calculated the ratio of the data from a single year of age to the total across the relevant 5-year age group for the different year or geographical scale and used it to partition the 2011 5-year age group data at ward level to single years of age. For England and Wales, single year of age data were available at Lower Super Output Area level for 2012 (population)²⁹ or 2014 (mortality);³⁰ data were aggregated for ward level before calculating the ratio to apply to the 2011 data). For Scotland, single year of age data were available for 2011, partitioned by local authority (population)³¹ or by country (mortality).^{32,33}

Total respiratory deaths for all ages, by local authority, were obtained from the Office for National Statistics (ONS) for 2011 (2013 for England; unpublished).^{34–36} We then split the local authority respiratory deaths for all ages into respiratory deaths by 5-year age groupings, using the 2011 ratios of each 5-year age group to total respiratory deaths for England, Wales (unpublished), and Scotland³² separately. Finally, we partitioned the respiratory deaths in each local authority into 1-year age groups using ratios of respiratory deaths in 1-year to

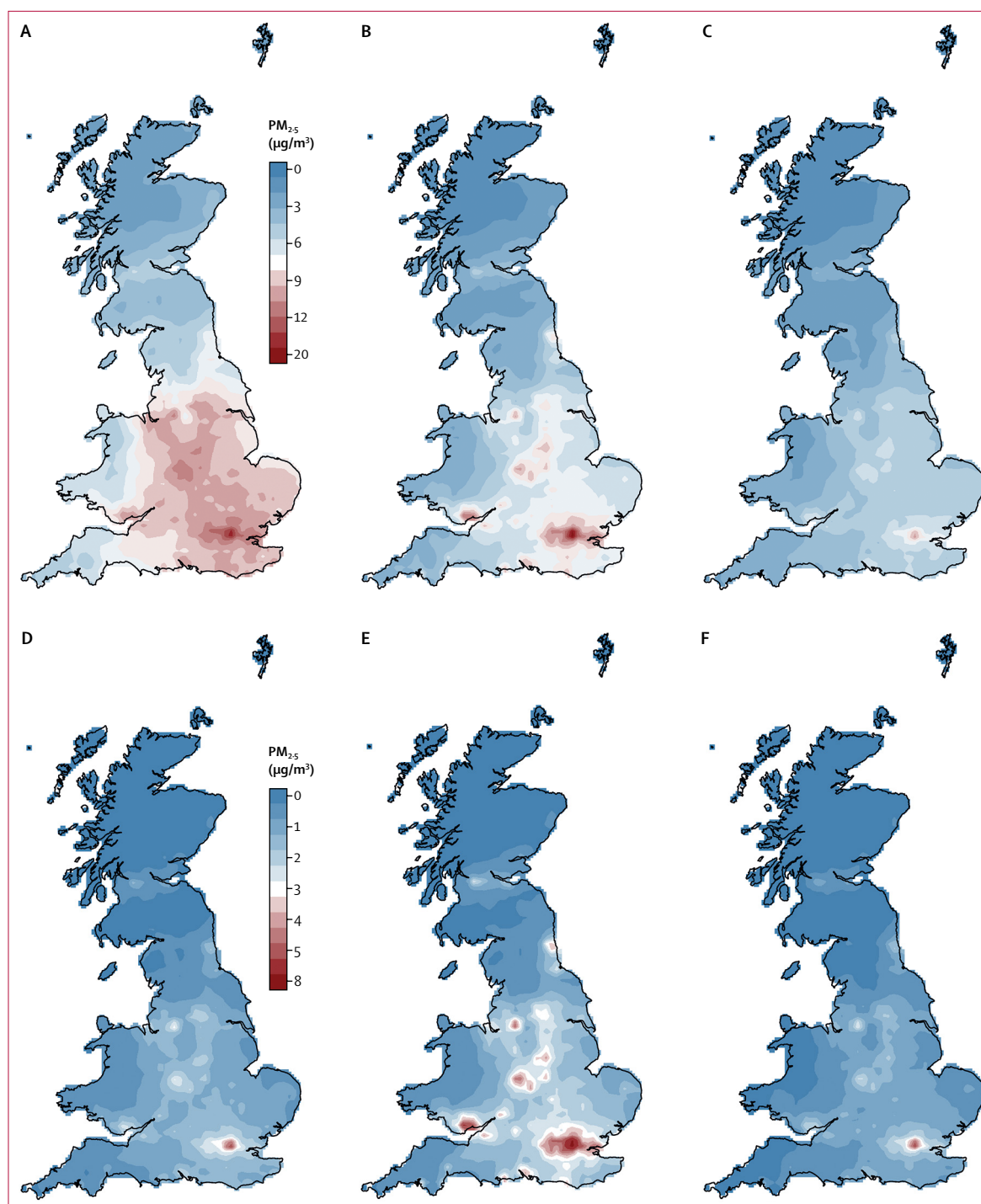


Figure 3: Annual $PM_{2.5}$ and primary $PM_{2.5}$ concentrations in Great Britain at 10 km resolution in the different scenarios

Data are mean, estimated by the summation of elemental carbon and organic aerosol. $PM_{2.5}$ concentrations for (A) 2011, (B) 2035 nuclear power scenario, (C) 2050 nuclear power scenario, and elemental carbon and organic aerosol data for (D) 2011, (E) 2035 nuclear power scenario, and (F) 2050 nuclear power scenario.

5-year age groups, using combined England and Wales data for 2015.³⁷

Birth projections by local authority were used until 2039; scaled from national birth projections for each

local authority by population until 2114, and kept constant thereafter.^{26,27,38,39} We requested all-cause mortality improvement projections by gender for all different ages from ONS.⁴⁰ These differed by year until 2038, and were

then held constant until 2154. We assumed that the relative proportions of respiratory deaths and other causes of death did not change over time because cause-specific mortality projections are not available. No account was taken of migration, given the diverse directions of the forecasts.

We chose concentration-response functions based on WHO⁴¹ or UK COMEAP⁴² recommendations because these were supported by a range of experts (appendix). The concentration-response function linking PM_{2.5} and mortality is based on the study by Pope and colleagues.⁴³ The summary estimate from 11 studies from Europe and North America in a 2013 meta-analysis⁴⁴ had the same hazard ratio⁴³ and was regarded as relatively robust in view of the WHO's report.⁴¹ The concentration-response function for NO₂ impacts on mortality are more uncertain. Although based on systematic reviews,^{44,45} difficulties exist when separating any independent effect of NO₂ compared with that of PM_{2.5}.⁶ For NO₂, we calculated impacts with and without a cutoff of 5 µg/m³. This cutoff was suggested by COMEAP⁴⁶ as representing the lower end of the range of concentrations used in the epidemiology studies on NO₂. We also calculated short-term impacts of NO₂ and PM_{2.5}, and will report these in a separate paper; however, we will report the results for

short-term impacts of ozone. We used the concentration-response function for short-term impacts of ozone on mortality that was recommended by COMEAP,⁴⁷ which had no threshold or cutoff and was based on a systematic review⁴⁸ of ten time-series studies covering 33 cities worldwide. We defined concentration-response functions (hazard ratios) per 10 µg/m³ and subsequently modified them on the logarithmic scale to give the new change in risk for the appropriate population-weighted mean concentration.

We assessed the differential exposure to air pollutants in different socioeconomic classes using the Carstairs index of deprivation (score and quintiles).⁴⁷ We derived and compared the modelled air quality at 10 km rural and 2 km urban scale, population-weighted for each ward, with Carstairs quintiles. The Carstairs index is a small-area composite indicator of relative socioeconomic deprivation, commonly used for epidemiological and health analysis.^{49,50} We calculated the Carstairs 2011 index, using 2011 census data, for each ward. We assumed the same deprivation patterns observed in 2011 for the 2035 and 2050 scenarios because relative deprivation patterns, such as deprivation quintiles, are fairly stable over time.⁵¹

Results

All three future scenarios saw electric vehicles comprising a large proportion of transport in Great Britain in the future (including hybrid, hydrogen, and compressed gas-fuelled vehicles). As such, concentrations of NO₂ in urban areas declined substantially (figure 2); although the baseline scenario showed large increases in NO_x emissions in some areas as a result of increases in combined heat and power sources.

The population-weighted concentrations of NO₂ decreased from 14.4 µg/m³ in 2011, to 11.3 µg/m³ in the baseline scenario in 2050. These concentrations decreased further to 7.1 µg/m³ (51%) and 5.7 µg/m³ (60%) in the nuclear power and low-greenhouse gas scenarios, respectively.

Total PM_{2.5} concentrations decreased by 2050 (figure 3) in all three scenarios. The population-weighted mean

	NO ₂ (µg/m ³)	Anthropogenic PM _{2.5} (µg/m ³)	Ozone in Great Britain (long term; µg/m ³)
2011	9.53–14.4	9.31	34.75
2035			
2035 baseline	2.44–6.98	4.74	28.43
2035 nuclear power scenario or low-greenhouse gas scenario	2.82–7.37	7.46	28.28
2050			
Baseline	6.43–11.25	5.77	26.77
Nuclear power scenario	2.55–7.08	5.36	28.04
Low-greenhouse gas scenario	1.33–5.65	5.20	28.10

Data are NO₂ (with and without a cutoff [counterfactual] of 5 µg/m³), anthropogenic PM_{2.5}, and ozone (from April to September the mean daily 8 h maximum ozone concentration was above 35 parts per billion [70 µg/m³]). NO₂=nitrogen dioxide.

Table 1: Population-weighted mean concentrations for pollutants in Great Britain

	PM _{2.5}	NO ₂ (no cutoff)	NO ₂ (5 µg/m ³ cutoff)	Ozone
Compared with 2011				
Baseline	17.839 (3.084 to 34.188)	6.488 (2.700 to 10.196)	6.406 (2.662 to 10.083)	1.126 (0.413 to 1.872)
Nuclear power scenario	16.716 (2.890 to 32.037)	11.575 (4.813 to 18.205)	11.298 (4.691 to 17.796)	0.909 (0.334 to 1.510)
Low-greenhouse gas scenario	17.367 (3.002 to 33.289)	13.666 (5.681 to 21.500)	13.223 (5.489 to 20.834)	0.885 (0.325 to 1.470)
Compared with baseline				
Nuclear power scenario	-1.122 (-0.194 to -2.151)	5.087 (2.113 to 8.010)	4.892 (2.029 to 7.713)	-0.217 (-0.079 to -0.362)
Low-greenhouse gas scenario	-0.472 (-0.082 to -0.900)	7.178 (2.981 to 11.304)	6.817 (2.827 to 10.751)	-0.241 (-0.088 to -0.403)

Data are life-years gained in millions (95% CI of concentration-response functions). Only effects for long-term exposure and mortality are shown and results might overlap, particularly for PM_{2.5} and NO₂. NO₂=nitrogen dioxide.

Table 2: Life-years gained (in millions) for the different scenarios in Great Britain

anthropogenic PM_{2.5} concentrations reduced from 9.3 µg/m³ in 2011, to 5.8 µg/m³ in the baseline scenario in 2050, to 5.4 µg/m³ (42%) and 5.2 µg/m³ (44%) in the nuclear power and low-greenhouse gas scenarios, respectively.

Crucially, the nuclear power (figure 3) and low-greenhouse gas (data not shown) scenarios both showed very large increases in domestic biomass burning and large increases in primary PM_{2.5} (about 75 000 tonnes per annum) compared with the baseline scenario (about 500 tonnes per annum). This source was distributed into 12 regions, according to UK regional wood use,⁵² and then within each region using a 1 km×1 km UK population, which resulted in increased PM_{2.5} concentrations (figure 3B) and elemental and organic aerosol data (figure 3E), therefore having important implications for public health.

Large projected increases in car and road freight mobility demand, accompanied with an absence of regulatory control of non-exhaust vehicle particulate matter (PM) emissions, lead to increased non-exhaust PM_{2.5} and PM₁₀ emissions. Consequently, non-exhaust emissions are likely to be the dominant source of primary PM from vehicles in the future, increasing PM₁₀ by about 15% compared with 2011 in the nuclear power scenario for example. PM_{2.5} road transport emissions are less affected by non-exhaust sources, and for the nuclear power and low-greenhouse gas scenarios, PM_{2.5} drops by 29% between 2011 and 2035, but by 2050 is predicted to increase by 17% compared with 2035, associated with increased non-exhaust emissions.

Ozone concentrations are projected to increase in winter because the NO_x removal process is reduced through reductions in NO_x emissions. So-called summer smog ozone concentrations are projected to decrease because of the reductions in emissions of ozone precursors. The population-weighted concentration reductions, compared with 2011, for NO₂, PM_{2.5}, and ozone in the three scenarios are shown in table 1.

We concentrated on comparison of impacts of policy scenarios. However, for context, maintaining concentrations of PM_{2.5} as unchanged at 2011 concentrations would lead to about 50 million life-years lost across the population in Great Britain over 2011–2154.

Reduced long-term exposures (2011–2154) to NO₂ in both the low-greenhouse gas and nuclear power scenarios lead to life-years saved (4 892 000 life-years for the nuclear power scenario [with cutoff] and 7 178 000 life-years for the low-greenhouse gas scenario [without cutoff]) for all-cause mortality rather than lost (table 2, figure 4). However, uncertainty exists over how much NO₂ itself contributes to the effects and how much is from other pollutants,⁶ such as PM_{2.5} or other traffic-related emissions; the associations that we used might overestimate the impact of NO₂. Because of this uncertainty, the decrease in the effects from NO₂ does not eliminate the concerns over the PM_{2.5} increase from biomass and non-exhaust emissions.

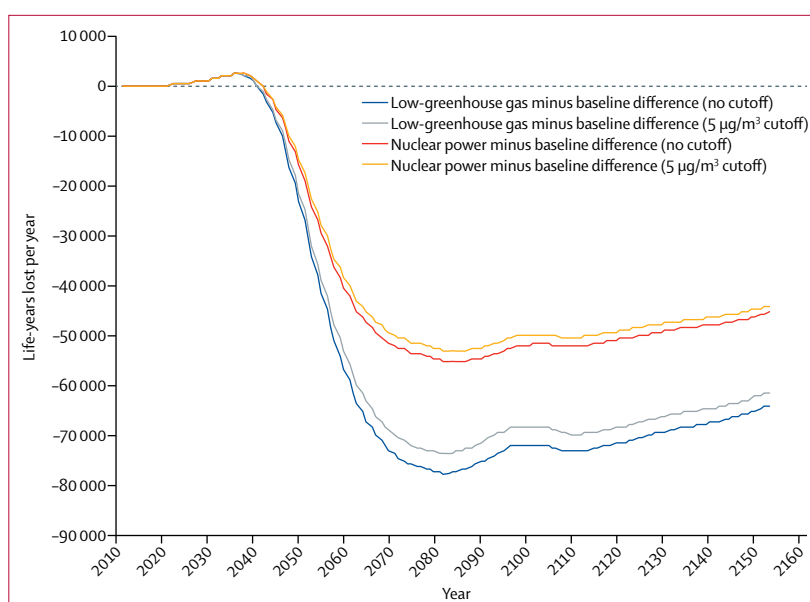


Figure 4: Changes in life-years due to NO₂ concentrations in low-greenhouse gas and nuclear power scenarios
We used 10 km rural per 2 km urban modelling, with relative risk 1.017 per 10 µg/m³, US Environmental Protection Agency lag, and Office for National Statistics birth and mortality projections. Negative life-years lost represents a decreased impact—ie, life-years gained. Ranges based on the confidence intervals around the concentration-response function recommended by the Committee on the Medical Effects of Air Pollutants in 2015,⁶ varied from 1.5 times the central estimate to 40% of it, and the ranges overlap to some extent between the two scenarios for most of the time period. NO₂=nitrogen dioxide.

The low-greenhouse gas scenario resulted in 472 000 more cumulative life-years lost and the nuclear power scenario resulted in 1 122 000 more cumulative life-years lost from long-term exposures to PM_{2.5} compared with the baseline scenario (table 2). This outcome represents an important lost opportunity for better health. It arises largely from the increase in biomass burning in these two scenarios, with the increase peaking in 2035. The life-years lost per year from long-term PM_{2.5} exposure illustrate the impact of the large increase in biomass (wood burning; figure 5). The life-years lost become negative (corresponding to life-years saved) only in the period of about 2060 and beyond (figure 5). These two scenarios that are compliant with the Climate Change Act would still result in around 4 months loss in life expectancy from birth in 2011. The analogous results for NO₂ (figure 4) show, because of the monotonic decrease in NO₂ concentrations until 2050, that life-years are saved throughout almost the whole period.

The increase in domestic biomass burning and in non-exhaust emissions mean that the reductions in total PM_{2.5} and PM₁₀ concentrations in the nuclear power and low-greenhouse gas scenarios are not as large as they might have been without the biomass increase. If the amount of primary PM_{2.5} projected to arise from domestic biomass burning was to be avoided, total PM_{2.5} concentrations in 2050 could fall even further than projected, down by roughly 50% in the highest areas in 2050 compared with about 25% reduction with the increased biomass use.

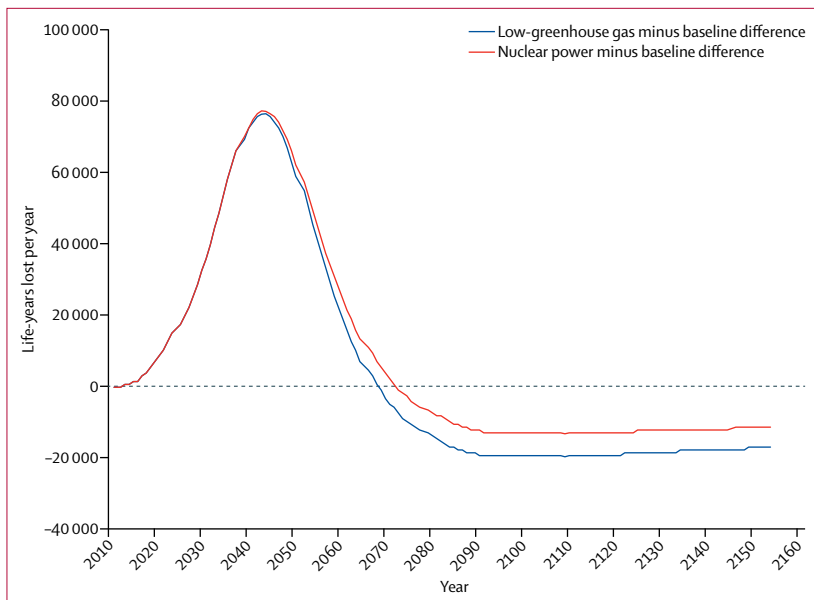


Figure 5: Changes in life-years due to anthropogenic PM_{2.5} concentrations in the low-greenhouse gas and nuclear power scenarios

We used 10 km rural per 2 km urban modelling, with relative risk 1.06 per 10 µg/m³; US Environmental Protection Agency lag, and Office for National Statistics birth and mortality rate projections. Ranges based on plausibility intervals from the Committee on the Medical Effects of Air Pollutants in 2010,²³ covering more than just statistical uncertainty, varied from twice the central estimate to a sixth of it, and the ranges overlap between the two scenarios.

We have not presented total changes in life-years associated with both PM_{2.5} and NO₂ combined because there is likely to be some degree of overlap that is poorly understood.⁶ The results for NO₂ are best regarded as an indicator of the effects of local scale reductions in the 2011 traffic pollution mixture (including effects of NO₂ itself), whereas the results for PM_{2.5} reflect changes in regional secondary particles, particle-dominated combustion, such as biomass burning with some contribution from traffic pollution too.

The long-term ozone exposure metric recommended by WHO⁴¹ is projected to decrease over time compared with 2011 for the baseline scenario, resulting in 1126 000 life-years gained compared with 2011 concentrations that remain unchanged (table 2). It also decreases for the other scenarios, but to a lesser degree. Thus, the low-greenhouse scenario resulted in 241 000 life-years lost and the nuclear power scenario resulted in 217 000 life-years lost compared with the baseline scenario (table 2). This outcome is a relatively small change compared with that for the other pollutants, due to the WHO threshold of 35 parts per billion and the effect being on respiratory mortality, not all-cause mortality. In contrast to the long-term metric, the important short-term ozone exposure metric, annual average of daily 8 h maximum ozone, is projected to increase in the baseline scenario leading to 25 500 more deaths brought forward (aggregated over 2011–50) relative to 2011 concentrations being maintained. Again, as for the long-term metric, the nuclear power and low-greenhouse gas scenarios, compared with the baseline scenario, have

worse consequences (a greater increase in concentrations with 3500 more deaths brought forward [range 1200–5900] for the nuclear power scenario and 5000 more deaths brought forward [1800–8500] for the low-greenhouse gas scenario; aggregated over 2011–50). However, the increased proportion of ozone in the mixture of oxidant gases, including NO₂, is potentially of some concern because ozone has a higher redox potential than does NO₂, and so could possibly increase the hazard from oxidative stress, although it is too early to be confident about this theory.

Regarding the social deprivation and inequality issues, in all scenarios, NO₂ and PM_{2.5} concentrations were significantly correlated with the deprivation score (Pearson's $r=0.309$ for NO₂ and $r=0.139$ for PM_{2.5}; $p<0.0001$; appendix). The correlations slightly decreased in 2050 compared with 2011 values for NO₂ but increased for PM_{2.5} (appendix).

Despite substantial reductions in air pollution concentrations (figure 3), the most socioeconomically deprived are still exposed to higher concentrations in 2050 (figure 6). Overall, NO₂ patterns by deprivation quintile suggest that as air pollution concentrations decrease, so does the ratio between mean concentrations in most deprived wards compared with the least deprived fifth of wards in 2011 across Great Britain (appendix). In 2011, mean concentrations in the most deprived wards were 4.3 µg/m³ higher than the least deprived (ratio 1.37; appendix). This difference decreased by 2050 in all scenarios—eg, to 2.8 µg/m³ in the baseline scenario, showing a narrowing in the air pollution inequality gap (ratio 1.31). This general pattern varies by region. In London, the inequality gap widens in the nuclear power scenario, with an NO₂ ratio of 1.45 in 2050, compared with 1.37 in 2011 (appendix). Similarly in Wales, where the ratio between mean NO₂ concentrations in most deprived compared with the least deprived fifth of wards in 2011 increases for all scenarios as NO₂ concentrations decrease (appendix). PM_{2.5} inequality patterns are similar across Great Britain and also Scotland, Wales, and London: as PM_{2.5} concentrations drop the ratio between the most deprived compared with the least deprived fifth wards in 2011 increases and the inequality gap widens, particularly for the nuclear power scenarios (appendix). In 2011, ozone concentrations are marginally higher in the least deprived compared with the most deprived fifth of wards, with the largest difference in London (ratio 0.93; appendix). For all scenarios, this difference disappears and concentrations are almost equal in Great Britain and separate regions assessed for 2050 (ratio between 0.98 and 1.00).

Discussion

Our study shows, through quantitative examples, that mitigation policies need to be carefully designed to avoid undue increases in harmful air pollution emissions. The effects on health of a changing climate and of policies to mitigate climate change are many and varied. The effects

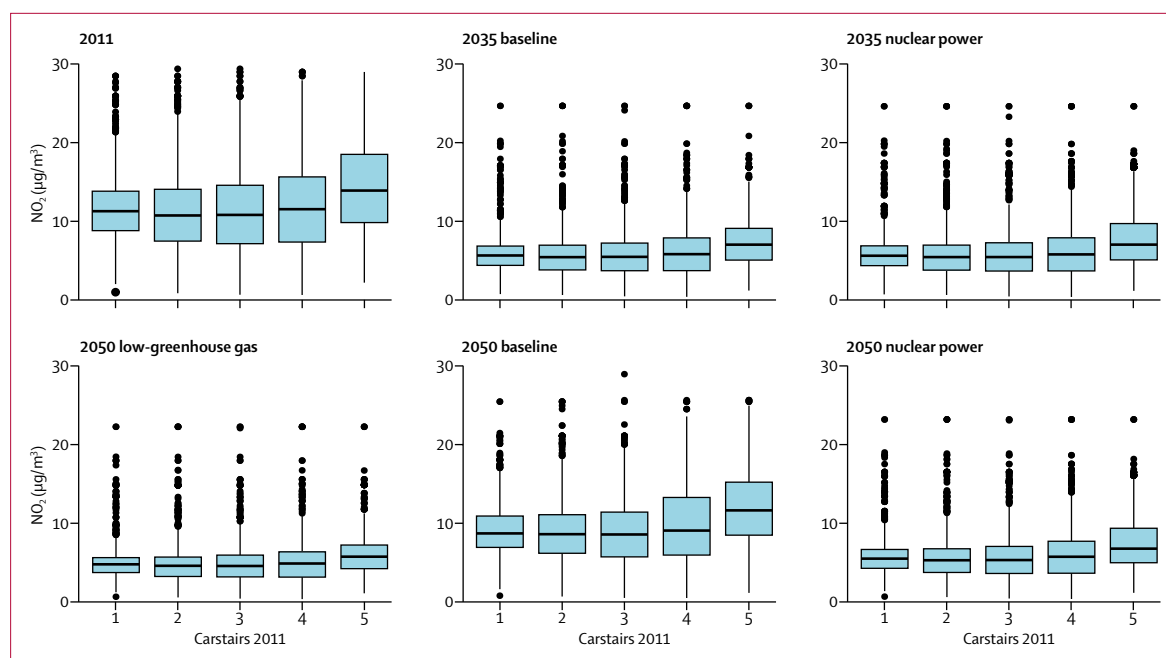


Figure 6: Distribution of ward 2011 level NO_2 concentrations in Great Britain

We used Carstairs 2011 deprivation quintile at 2011 baseline and under the different scenarios. NO_2 =nitrogen dioxide. 1=least deprived. 5=most deprived.

of mitigation policies have the potential to make dramatic improvements in public health through their parallel improvements in air quality. Modelling studies, such as ours, are needed to assess fully the public health impacts of the air quality changes arising from climate change interventions and to inform decisions on optimal policy choices.

Concentration-response functions for use in benefit assessment of population-based policies are usually based on effects in the general population (because exposure is reduced for everyone) and on all-cause mortality (to avoid misdiagnosis). However, underlying these general public health impacts will be increased effects in susceptible groups, such as myocardial infarction survivors,^{52–54} and effects on cause-specific mortality, including mortality from ischaemic heart disease, chronic obstructive pulmonary disorder, and lung cancer.⁷

This paper has concentrated on mortality impacts—these would dominate the monetised benefits in a cost-benefit analysis.⁵⁵ But morbidity impacts are still important in terms of effects on quality of life and health-care costs and increases and decreases are likely to occur in line with the direction of effects on mortality in respiratory and cardiovascular hospital admissions, respiratory symptoms, asthma outcomes, restricted activity days, and other outcomes with varying degrees of evidence behind them. We will report these in a future publication.

With all projections, many uncertainties exist. We included both mortality projections and birth projections, which is a strength compared with leaving these constant. However, many other changes in health outcomes and risk

factors affecting these outcomes over time are possible. We only included these to the extent that projections in mortality are extrapolated from past improvements that reflect, for example, improvements in medical treatment. Climate change itself might change patterns of disease and, thus, susceptibility to air pollution. These issues need to be kept in mind when interpreting the results.

This work has developed a model system that has the explicit assessment of the potential benefits of climate change mitigation interventions, with a state-of-the-art energy system and air quality and health impact models. At present the system is configured specifically to Great Britain but could be extended fairly straightforwardly to other countries or regions. Few future energy scenarios have been analysed for Great Britain, but this analysis has produced results that are important for the future development of coordinated policies to mitigate both climate change and air pollution. Our results suggest that scope is considerable for such coordination to maximise the substantial potential benefits to public health which could be achieved. Furthermore, although this paper has concentrated on air quality changes, analysis of other co-benefits of climate change policies, such as reductions in noise or increases in physical activity, could be integrated with our system within the life table analysis. Notably, the health impacts of a changing climate per se have not been covered in this work because the intention was to compare different routes to the same amount of greenhouse gas emission reductions. One limitation of our work has been the fact that in developing the model we have only been able to investigate some illustrative future scenarios. Also,

the absence of projected rates for ethnicity over time meant that we were only able to consider deprivation impacts and not impacts on ethnic minority groups or combined ethnicity-deprivation impacts.

In air quality terms, the reductions in NO₂ concentrations are extensive, and although uncertainty exists over the quantification of the impact of this pollutant, the health effects from NO₂ exposure (or other closely correlated traffic combustion pollutants, also reduced by fleet electrification) should decrease significantly.

The situation for PM_{2.5} is to some extent clearer in terms of quantification of the health impacts in that there is more confidence for the concentration-response functions. However, the change in the mix of future emissions strengthens the need for studies to investigate, and health impact methods to account for, the differential toxicity of PM components. Although total PM_{2.5} concentrations are projected to decrease mainly because of reductions in exhaust emissions from vehicles and from reductions in the precursors of secondary particles, domestic wood burning PM_{2.5} and non-exhaust emissions from road vehicles are projected to increase by relatively large amounts. For example, the primary combustion from wood burning, which contains a human carcinogen (polycyclic aromatic hydrocarbons), and non-exhaust PM, has been suggested from some initial research to cause potentially adverse health effects.⁵⁶

Assessing pollutant mixtures is also important, with short-term exposure to ozone remaining a potential problem. Studies on the effects of long-term exposure to ozone are contradictory and COMEAP⁴² did not recommend quantification. WHO⁴¹ did recommend quantification but only with a threshold that means effects are relatively small compared with PM_{2.5}. Any new evidence changing the balance of the effects of long-term and short-term exposure to ozone would have considerable policy implications. The degree of overlap between NO₂ and other pollutants is crucial in determining net benefits. The opposing outcomes of NO₂ and some PM_{2.5} components over time might assist in distinguishing pollutant impacts in epidemiological studies in the future.

The future scenarios for NO₂ indicate a possible reduction in the differences between most and least deprived populations as measured by the ratios, apart from the high biomass 2035 nuclear power scenario. This scenario is less clear for PM_{2.5}, for which the ratios are indicated to possibly increase. Despite the overall reduction in absolute concentrations of air pollution exposure across the spectrum of deprivation, differences in exposure between the most and least deprived populations remain in all future scenarios and for all pollutants. This situation is assuming that patterns of residence by deprivation classes remain broadly the same as for 2011. Addressing these persistent socioeconomic differentials must be a core component of climate change and air pollution policies if they are to maximise their potential for long-term public health benefits.

This research has provided some key messages for future climate and air quality policies and has produced a tool for investigating optimal pathways to the Climate Change Act target, which achieve climate change policy goals and minimise the impacts on public health and the wider environment. If alternative scenarios involving substantial investment in non-biomass energy sources were to be considered, their impacts could be modelled using the tool we have developed here. Policy opportunities to achieve healthier environments are clear, but challenges remain to improve both environment and health for the most vulnerable groups in our societies.

Contributors

MLW was responsible for the overall concept and design of the work. MCL and SP ran the energy systems model and developed the scenarios in conjunction with MLW and SDB. SDB produced the emission inventory from the UK TIMES model. SDB and NK ran the air pollution model. DD and HW calculated the health impacts, MBT and DF carried out the social deprivation analysis, and MH did the economic analysis. MLW undertook the initial writing of the paper with contributions from all co-authors.

Declaration of interests

MLW and HW report grants from the National Institute for Health Research, during the conduct of the study. MCL reports grants from BHP Billiton, outside the submitted work. All other authors have no competing interests.

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